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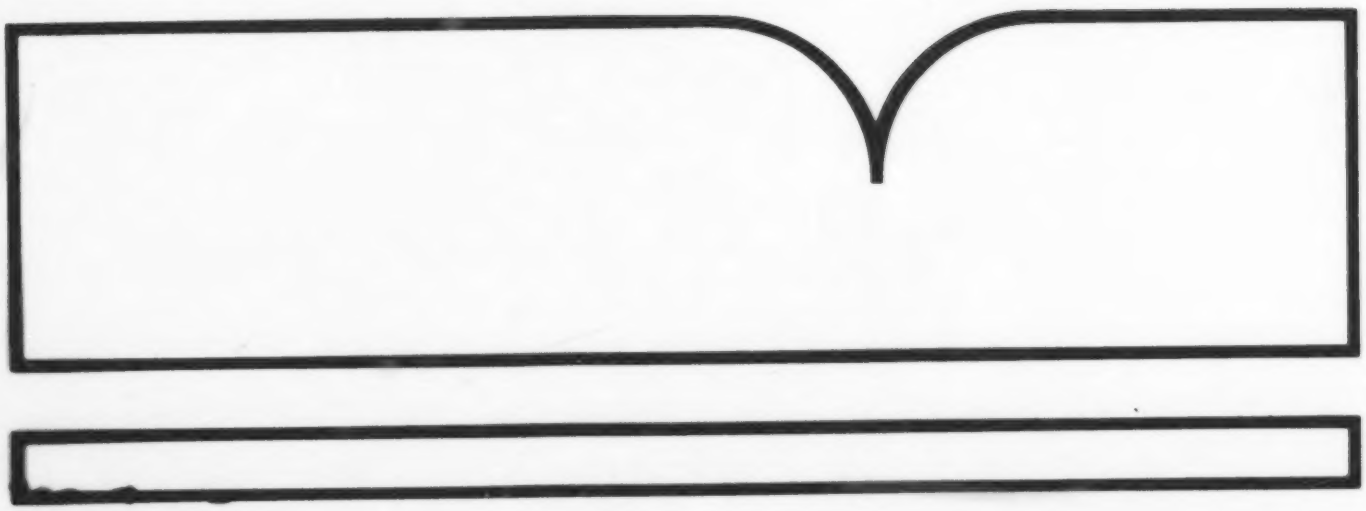

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Analysis of National Water Pollution Control Policies
1. A National Network Model

LEONARD P. GIANESSI AND HENRY M. PESKIN

Resources for the Future, Inc., Washington, D.C. 20036

G. K. YOUNG

GKY and Associates, Inc., Springfield, Virginia 22151

In order to study the national implications of federal water pollution control policies, the authors developed a water pollution network model. This model links both point and nonpoint sources of pollution at the county level of detail with a national network of major rivers, lakes, reservoirs, and bays. While the geographical scope of the model and existing data availability required making many simplifying assumptions, the model permits comprehensive analyses that are not possible with existing techniques.

INTRODUCTION

There have been at least three efforts to model water pollution for the entire nation. First there was the study of *Wollman and Bonem* [1971], followed by the National Residuals Discharge Inventory [*Luken et al.*, 1976] prepared for the National Academy of Sciences, and the Environmental Protection Agency's SEAS model [*House*, 1977]. In each case these efforts attempted to describe the aggregate water flow through a relatively large geographic region. Generally, these conform to the aggregated subareas (ASA's) of the Water Resource Regions as defined by the *U. S. Water Resources Council* [1970]. The exact linkages between rivers, reservoirs, lakes, and bays within the region were not specified.

While this crude approach may have served adequately the objectives of the model builders, our research goals demand certain refinements. First, it is necessary that the model identify specific rivers, lakes, etc., since estimates of the quality of these bodies of water are crucial to an assessment of the impacts of pollution control policy. Next, it is necessary that points along these bodies of water be associated with waterborne discharges originating in specific upstream and local counties, since our point and nonpoint data on discharges are at the county level of detail. Finally, it is necessary that the coverage of the model be national in scope, since one of our objectives is the analysis of the distribution of policy costs and benefits throughout the United States.

Unfortunately, a geographically aggregative approach, such as used in the *Wollman and Bonem* [1971] study, fails to provide the necessary specificity. If, for example, a large region contains a fast-flowing river in a rural area and a slow-moving river in a heavily industrialized area, the aggregative approach, in effect, assumes a single river with a flow equal to the sum of the flows of the two rivers but with unknown velocity characteristics. The implicit average water quality of this single abstract river could be far better than the quality actually enjoyed by those living along the slow, industrialized river or far worse than the quality actually enjoyed by those living along the swift, rural river. The potential for a misleading description of actual water quality is even greater if either of the two rivers is interrrupted by natural or man-made lakes and reservoirs. The aggregative approach fails to account for the fact that these lakes may trap certain waterborne residuals and thus improve downstream water quality. In addition, because the aggregative approach does not consider transport and degradation of pollutants, the water quality is implicitly assumed uniform throughout the large regions.

While the aggregative approach has obvious deficiencies, the alternative of building highly detailed models for specific water bodies is prohibitively expensive if the study is to be national in scope. Indeed, the cost of modelling a single river at the level of detail commonly used for hydrologic studies could easily exceed the budget of our entire project.

The approach discussed in this paper thus represents a compromise between highly aggregative modelling and detailed hydrologic modeling [*Spofford et al.*, 1976; *Spofford*, 1973; *Thomann*, 1972].

OVERVIEW OF THE MODEL

The water model basically consists of a water network, a set of assignment functions for injecting pollutants into the network, and a system of equations or rules for transmitting the pollutants throughout the network.

Water Network Design

The basic network design relied on judgments concerning the importance of bodies of water in terms of their size, their role as receptors of pollutants, and whether they receive runoff from reasonably large drainage areas. For example, an effort was made to include all rivers with average flows at their mouths exceeding 1000 $ft^3\ s^{-1}$ (28 m^3/s). However, because of their lack of importance as pollutant and runoff receptors, seven rivers in the Northwest were excluded even though their average flows exceed 4000 $ft^3\ s^{-1}$ (112 m^3/s).

On the other hand, relatively small rivers, such as the Charles in Massachusetts or Ohio's Cuyahoga River, are included becuase they are receptors of pollutants from major urban areas. Similarly, a number of very small rivers in arid Western areas are also included because they receive runoff from two or more counties.

Bays, lakes, and segments of ocean shorelines are also included if they meet the above three criteria. The same statement is true for major reservoirs. However, any reservoir, regardless of whether it meets the criteria, is also included if it interrupts the flow of a river in the network.

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Fig. 1. Map of rivers, lakes, and reservoirs included in the national water quality model.

Figure 1 is a map showing the location of the rivers, lakes, and reservoirs included in the model. In all, the network of bodies of water contains 304 rivers, 175 lakes and reservoirs, 37 bays, 10 segments of Great Lakes shoreline, and 26 ocean shoreline segments. The size distribution of the rivers is shown in Table 1.

Obviously, many short tributaries and streams have been excluded from the network even though they may be receptors of pollutants. The implicit assumption of the model is that the travel time between these short tributaries and the main river is negligible, with the result that their pollutants arrive intact.

In addition to selecting bodies of water for the network, it was necessary to segment them in order to locate the network's nodes. Points on rivers were selected as nodes if any of the following criteria were met:

1. The point is near a major population center.
2. The point is where a major tributary enters.
3. The point is the beginning of an estuary.
4. The point is the entrance to a reservoir.
5. The point is the river's mouth.

Additional nodes were established if the distance between the points selected according to the above criteria became very large and the river had passed through four or more counties. In all, 1051 river nodes were established (along with an additional 249 nodular points along bays, lakes, and ocean shorelines). The average distance between nodes is 66 miles (106 km). Since the network's average river is 165 miles (265 km) long, there are about 3.5 nodes per river, although a major river, such as the Mississippi, obviously has many more.

In general, in order to ease data development problems, the precise location of a node was made to coincide with a United States Geological Survey (USGS) long-term streamflow measuring station. Unfortunately, for about 40% of the nodes this was not possible.

Each county in the United States (except for 13 completely arid counties) is assigned to at least one node. In certain cases, more than one node was assigned, if it was clear that the county's drainage went into more than one body of water in the network. The assignment was based primarily on inspection of the runoff maps of the *U.S. Department of Agriculture, Soil Conservation Service* [1970]. However, often major municipal sources of pollutants are channeled into waterbodies that would not be the receptors if natural drainage patterns were followed. Where knowledge of such situations existed, the runoff maps were ignored. Thus, in Chicago, the Chicago River, whose natural direction of flow has been reversed, was made a receptor of point source pollutants rather than Lake Michigan.

While the resulting network is quite large, it is not totally connected. In fact, 44 independent subnetworks can be defined, although often the river-lake-reservoir-bay linkages within a given subnetwork can be quite complex. The Mississippi subnetwork, which contains 124 rivers and 78 lakes and reservoirs, is perhaps an obvious example.

Pollution Assignment Functions

The purpose of the pollution assignment functions is to de-

TABLE 1. Size Distribution of Rivers included in the Model

CFS	m^3/s	Number
≤999	≤27	70
1000–3999	28–111	109
4000–6999	112–195	40
7000–9999	196–279	27
10000–19999	280–559	30
20000–49999	560–1399	13
50000–99999	1400–2799	6
≥100000	≥2800	9

Freshwater streamflow at mouth.

termine the amount of pollutants inserted into the network at or between each node. The model recognizes that certain activities lead to a direct insertion of pollutants into bodies of water. These activities are referred to as point sources. Other activities lead to an indirect insertion, and these are referred to as nonpoint sources. Still other activities lead to both direct and indirect insertions.

Examples of point sources are most industrial activities, as well as municipal treatment. Examples of nonpoint sources are most agricultural activities and urban runoff, and examples of both point and nonpoint sources are mining activities.

The model assumes that all pollutants coming from point sources (as well as the point source component of mining activities) are injected into the network at nodal points. The model also assumes that most nonpoint source pollutants are injected uniformly between the nodes. However, urban runoff is injected at nodal points. In particular, the model aggregates the point source pollutants and urban runoff in all counties assigned to a particular node. These are assumed inserted at the same node. The counties' nonpoint source pollutants (other than urban runoff) are assigned to the segment between this node and the next downstream node.

Estimates of pollutant discharge from point source sectors have been made for 1972 levels of control at the county level of detail. Whenever possible these estimates are based on survey data of actual discharges at individual plants in 1972. The basic approach for most point source sectors has been to assemble a plant inventory showing county location and plant output for all plants in the industry in 1972. Waste coefficients developed by the Environmental Protection Agency are then used to estimate total waste generated at the plant in 1972 assuming no end-of-pipe pollution control. Examination of the actual end-of-pipe wastewater treatment equipment in place in 1972 is then made for each plant wherever possible. Standard removal efficiencies associated with the equipment are assumed to reduce the generated waste levels to the actual discharge level for 1972. The basic source of data used in this project has been a set of technical reports known as Development Documents prepared by the *U.S. Environmental Protection Agency* [1974].

For sectors where complete plant inventories could not be assembled a cruder method of assigning sector by sector pollutant estimates at the county level has been employed. For these sectors a national discharge estimate is made by assuming that all plants employ 'standard' production processes, generate similar amounts of waste per unit of output, and employ some 'average' level of pollutant removal efficiency. The assignment of this national discharge estimate to individual counties is made in proportion to the county's employment in the industry. Estimates of employment by county are derived from the Public Use sample of the 1970 census. A detailed description of the methods used in making the estimates for all sectors is available in working papers maintained by the authors (L. P. Gianessi, unpublished data, 1978).

Assignment to nodes of point source pollutants is straightforward: the point source pollutant estimates for all counties assigned to a node are added together. Both annual and daily estimates of point source pollutant discharge are made. (The daily estimates are equal the annual estimates divided by 365.)

The assignment of nonpoint source pollutants to nodes is more complex, since there is no direct linkage between generation and insertion into a waterbody. Rainfall and surface conditions play a large role. The effect of these conditions must be modeled.

Depending on the particular nonpoint source in question, we approached this problem in one of three ways. For certain sectors (e.g., nonirrigated cropland, pastureland, irrigation return flow, urban runoff, stream bank erosion, and acid mine drainage) we relied on the analyses of other investigators, taking their estimates of annual pollutant discharge as given [*Midwest Research Institute*, 1975; *Appalachian Regional Commission*, 1969; *Heaney*, 1977; *Toups Corporation*, 1975; and *Wade and Heady*, 1976]. Often, however, their estimates were defined for geographical regions other than counties. In these cases we had to prorate the estimates to counties, sometimes rather arbitrarily. For a second set of sectors (e.g., ship ballast cleaning, accidental oil spills, recreational boating, and dumping of sewage sludge in oceans), national estimates of discharges have been prorated to counties using detailed information on the location and extent of the sector activities.

Finally, for a third set of sectors (e.g., sediment from construction, forestry, and mining), national estimates of discharge have been prorated to counties according to the county's share of employment in these activities weighted by an estimate of runoff per acre. In this manner, effects of terrain and rainfall were crudely accounted for. Thus, if national discharge of the sector is N, county i's share of the employment of the sector is E_i, and runoff per acre in i is R_i, then the estimate of discharge in i, N_i, is

$$N_i = \frac{E_i R_i}{\sum E_i R_i} N$$

It is also necessary to prorate estimates of annual discharges of nonpoint sources to seasons. Obviously, this proration can not be done by assuming a uniform daily discharge since rainfall and runoff vary throughout the year. Rather, seasonal streamflow data for each body of water point for the year under analysis (1972) is first estimated. (The sources of these data are described below.) It is then assumed that runoff will be proportional to streamflow. The annual nonpoint source pollutants are prorated to the seasons according to the season's share of annual runoff. Daily seasonal estimates are computed by dividing the seasonal total by 91.25. For estimating runoff to nonriver bodies of water, seasonal streamflow shares of an associated river point are used.

One final adjustment is required for all the nonpoint pollutant estimates for the year under analysis. The annual estimates of pollutant discharge from nonpoint sources were computed using long-term data on rainfall and runoff. Thus the pollutant estimates are for a 'typical' year. Since our streamflow estimates are for a specific year, it is necessary to calibrate the nonpoint pollutant estimates for each body of water according to the ratio of the specific year's streamflow (1972) to the mean long-term streamflow.

Transport Function Data Development

Data describing the hydrodynamic characteristics of the water network were assembled in order to allow us to use the pollution transport functions outlined in the next section. Data on several variables had to be developed for each node in the system. Since groups of variables come from similar sources (e.g., various USGS publications), the discussion will focus on these groups rather than on individual variables.

Generally, values for flow, altitude, and distance were

drawn from USGS publications. In particular, long-term average flows for about 800 upstream nodes came from a certain work by the *U.S. Geological Survey* [1975]. If long-term average flows were not available, the average flow for 1970 was used. The flows of the mouths of rivers discharging into the ocean came from *Bue* [1970] and *Iseri and Langbein* [1974]. However, flows for rivers connecting the Great Lakes did not come from USGS publications but rather from the *International Great Lakes Level Board* [1973]. Average seasonal flows and the lowest daily flow for the year under analysis (1972) came from the *U.S. Geological Survey*, [1972, 1973]. The seasons are defined as Winter (December–February), Spring (March–May), Summer (June–August), and Fall (September–November).

If the USGS did not provide a flow estimate for a certain node, the long-term average flow was estimated either by interpolating between the flows above and below the node or by using the same flow as above or below the node and adjusting the number for any known tributary inflows. The seasonal and lowest daily flow estimates for these nodes were assumed to be in the same ratio as the seasonal flow of the nearest node (for which data were available) to that node's estimated long-term flow.

Data on altitudes, from *Gannett* [1906], are available for most nodes. For the remaining nodes the altitude of a nearby city was used.

Distance between nodes for most nodes corresponding to USGS stations came from the *U.S. Geological Survey*, [1975]. Other distances were estimated either from the atlases of the *U.S. Department of Agriculture, Soil Conservation Service* [1970] or *Rand McNally* [1976]. The information on altitudes and distance between points allowed us to calculate the average slope for each segment.

Regional offices of the USGS make approximately ten measurements per year on width, cross sectional area, and velocity at all gauging stations of which about 600 stations correspond to our nodes. These observations are hand-recorded on USGS Form 9-207. By examining these forms, covering a 15-year period, our flow estimates could be closely bracketed between one higher and one lower measured flow. The associated width, velocity, and area measurements for these flows were interpolated to estimate velocities, widths, and areas for the average long-term and seasonal flow estimates. (Depth was estimated simply by dividing area by width.)

The interpolated width and area estimates were constrained not to decrease as flow increased, even though apparent decreases were recorded on the 9-207 forms. Velocity estimates realistically could either decrease or increase as flow increases. However, velocity was not allowed to decrease with increased flows if estimated area and width were constant.

For those nodes not corresponding to the 600 USGS stations, width and depth estimates were made by least squares extrapolation using about 300 of the possible 600 observations as the data base. Specifically, regressions of the following form were fitted with the data;

$$\log W = \alpha_1 + \beta_1 \log Q + \gamma_1 \log L$$

$$\log A = \alpha_2 + \beta_2 \log Q + \gamma_2 \log S$$

where W is width (in feet), A is area (in ft^2), Q is flow (in ft^3/s), L is length from river's source (in miles), and S is the river's slope (in ft/mile). (Nodes with nonzero values for all variables must be used for the regressions. Only 300 of the 600 nodes qualified.) Velocities were determined by dividing Q by A. The results are as follows: (standard errors are in parentheses.)

$$\log W = \underset{(0.13540)}{2.3117} + \underset{(0.01361)}{0.41586} \log Q + \underset{(0.01411)}{0.01119} \log L \quad \bar{R}^2 = 0.78539$$

$$\log A = \underset{(0.10937)}{0.81539} + \underset{(0.01052)}{0.82486} \log Q - \underset{(-0.01952)}{0.14315} \log S \quad \bar{R}^2 = 0.94752$$

Long-term lake and reservoir volumes were set equal to lake and reservoir capacities. Seasonal data on volumes for 1972 for 132 lakes and reservoirs in the network were developed from information in works by the *U.S. Geological Survey* [1979 and 1973]. Estimates of seasonal volumes for the remaining lakes and reservoirs were made by first calculating the ratios between capacity and seasonal volumes for the above 132 lakes. The average value of these ratios for each season was then multiplied by the capacities of the remaining lakes and reservoirs to estimate their seasonal volumes. A similar procedure was used in estimating low volumes for lakes and reservoirs. Eight lake sinks in the West, which were known to be completely dry in 1972, were assigned zero seasonal values.

Most lakes and reservoirs in the model are connected to rivers which are also included in the model. For these lakes, the seasonal outflow estimates are the same as the seasonal streamflow estimate for their downstream river connection. However, there are twelve lakes in the West which are sinks in the model. Their seasonal and low outflows are entered as zeroes. Finally, there are 44 lakes in the model which are neither sinks nor connected to a downstream river point in the model. They are usually connected to another lake or reservoir in the model, and for modeling purposes the connecting river segment is ignored. For these lakes and reservoirs the connecting river segment was identified and its seasonal streamflow was determined from USGS records [*U.S. Geological Survey*, 1972, 1973].

Maximum tidal velocities (in knots) are drawn from National Ocean Survey data [*National Ocean and Atmospheric Administration*, 1978], the average of the reported flood and ebb values. The National Ocean Survey reported these values for 55 of our 107 estuaries. The remaining estuaries were assigned a maximum tidal velocity (1.15 knots: 2.13 km/hr) equal to the average of the 55 observations. Other estimates made for estuaries include the incoming freshwater streamflows (seasonal and low flows), the velocity of these freshwater flows, and the depths. These data are drawn from the streamflow, velocity, and cross section estimates made for those river nodal points designated as estuaries. The estimates for nonriver estuaries are averages of the data for the river sections.

Surface temperature by season for each node was estimated by noting in which temperature zone (defined as the region between isotherms of 5° difference) the node was located as delineated in the atlas of the *Water Information Center* [1973]. The temperature assigned to the zone was the temperature as recorded by an arbitrarily selected USGS measuring station in the zone. These recorded temperatures are found in works by the *U.S. Geological Survey*, 1972, 1973].

Pollution Transport Functions

The purpose of the pollution transport functions is to con-

vert injections of pollutants at the nodal points to concentrations of pollutants at and between the nodes. In general, the process described by all the functions is similar: initial injections of pollutants at a node are mixed with quantities of pollutants arriving from upstream nodes, the mixture is diluted by the flow at the node, and the mixture is transported to the next node while, on the way, certain constituents are removed by natural processes.

The model assumes that dissolved solids (DS) are totally conserved. Therefore, the concentration of this constituent at node j is:

$$C_j^{DS} = \left(\sum_{k=1}^{j-1} C_k^{DS} q_k + W_j^{DS} \right) / q_j \tag{1}$$

where C_k^{DS} is the concentration at node k, q_k is the flow at the kth node, and W_j^{DS} is the input of dissolved solids at the jth node. This formula is applied sequentually to all nodes, moving from the tributaries to the main stems in each of the 44 independent water basins that make up the entire network. Concentrations are conventionally measured in milligrams per liter.

The concentration estimates for lakes (and bays in the Great Lakes regions) are made by dividing the DS mass by seasonal outflow estimates. DS concentration estimates are made neither for bays located on the oceans nor for the oceans themselves. However, DS concentration estimates are made for estuaries located on rivers discharging into the oceans.

For those nodes of bodies of water in the model that have two downstream connecting points the amount of DS assigned to each of the downstream points is prorated in proportion to the flow in the downstream segments. This proportion is also maintained for the substances that attenuate (discussed next).

Providing they originate from point sources or nonpoint urban runoff, the concentration at the $j+1$th node of substances that attenuate (e.g., biochemical oxygen demand (BOD_5), ultimate oxygen demand (UOD), total phosphorus (TP), and total Kjeldahl nitrogen (TKN)) is calculated by the following:

$$C_{j+1} = \frac{W_{j+1}}{q_{j+1}} = \frac{W_j e^{-K_1 \tau}}{q_{j+1}} + \frac{W_{j+1}^{\,0}}{q_{j+1}} \tag{2}$$

where C_{j+1} is the concentration at the $j+1$th node, W_j and W_{j+1} are pollutant inputs at j and $j+1$, respectively, $W_{j+1}^{\,0}$ is the 'initial' pollution input at $j+1$ due to insertions of the attenuated substance at node $j+1$, and τ is the travel time between nodes j and $j+1$. Travel time is the product of the segment's length times the stream velocity.

The attenuation rate K_1 differs by substance: 0.1 (per day) for phosphorus, 0.2 (per day) for TKN, 0.6 (per day) for BOD_5 and UOD if the concentration is greater than 10 mg/l, 0.4 (per day) for BOD_5 and UOD if the concentration is between 3 and 10 mg/l, and 0.2 (per day) for BOD_5 and UOD if the concentration is less than 3 mg/l. With the exception of TP, these values are assumed valid only at 20° C. Thus they are temperature corrected according to the formula [*Thomann*, 1972]:

$$K_1 = (K_1)_{20} [\theta^{(\text{Temp}-20)}]$$

where θ is 1.047, where Temp is measured in °C, and where $(K_1)_{20}$ are the above values of K_1.

For lake and reservoir reaches the attentuation coefficients K_1 are reduced to one half their river values in order to account for quiescence. Effective lake flow through time is assumed to be reduced by one half in order to account for stratification. Lake travel time is calculated as lake volume divided by outflow.

We assume that pollutants originating from agricultural nonpoint sources are evenly injected along the reach between nodes j and $j+1$. That is, if W_j is the amount of nonpoint pollutant assigned to node j and $X_{j,j+1}$ is the distance between node j and $j+1$, we define the distributed load as

$$W_{j,j+1} = W_j / X_{j,j+1}$$

Formula (2), altered for nonpoint source pollutants that attenuate, becomes

$$C_{j+1} = C_j e^{-K_1 \tau} + \frac{W_{j,j+1}}{K_1 A_j} (1 - e^{-K_1 \tau}) \tag{3}$$

where A_j is the cross section of the reach at node j.

A particular application of formula (2) is for the purpose of estimating nodal concentrations of UOD. This is a first step in calculating a dissolved oxygen curve between nodes as it relates to point sources of oxygen demanding pollutants (BOD_5 and TKN). (The dissolved oxygen curve related to nonpoint sources of these pollutants is calculated in a slightly different manner.)

UOD is commonly assumed to equal 1.5 BOD_5 + 4.57 TKN. The 1.5 factor is a typical assumption for converting 5-day BOD at 20°C to carbonaceous UOD, and the 4.57 factor is based on the stochiometry of converting ammonia to nitrate. The UOD concentration at node j, C_j^{UOD}, is determined by formula (2). The dissolved oxygen concentration, $\phi(X)$, at any distance X along a river reach between j and $j+1$, the next downstream node, is determined by the Streeter-Phelps formula [*Streeter and Phelps*, 1925]:

$$\phi(X) = \phi_s - \left\{ \frac{K_1 C_j^{\text{UOD}}}{K_a - K_1} (e^{-K_1 \tau(X)} - e^{-K_a \tau(X)}) + (\phi_s - \phi_0) e^{-K_a \tau(X)} \right\} \tag{4}$$

where ϕ_s is the saturation level of dissolved oxygen (in mg/l) estimated as $\phi_s = 14.55 - 0.382(\text{Temp}) + 0.00543(\text{Temp})^2$, where, as before, Temp is temperature in °C. This relationship is based on quadratic regression fitted with data from *Fair and Geyer* [1954]. The initial dissolved oxygen, ϕ_0, at the beginning of the subnetwork is assumed to be at 90% of saturation, or $\phi_0 = 0.9\ \phi_s$. The reaeration coefficient K_a is estimated by $K_a = 7.6\ V_j / h_j^{1.33}\ 1.024^{(\text{Temp}-20)}$, where V_j is the stream velocity (in ft/s) at j and h_j is the mean depth (in feet) at j. This approximation of K_a is found in the work by *Langbein and Durum* [1967]. K_1 is the same parameter as in formula (2). As with formula (2), K_1 is temperature-adjusted. Lastly, $\tau(X)$ is the travel time from the node to any downstream X, or $\tau(X) = X/V_j$.

Formula (4), like formula (2), is modified for agricultural nonpoint sources of UOD. As before, agricultural nonpoint UOD will be assumed to be evenly injected along the reach between nodes j and $j+1$ at a rate equal to W_j^{UOD}. The dissolved oxygen concentration due to nonpoint UOD under this definition of $W_{j,j+1}^{\text{UOD}}$ is

$$\phi(X)_{\text{nonpoint}} = \phi_s - (\phi_s - \phi_0) e^{-K_a \tau(X)} - (1 - e^{-K_a \tau(X)}) \frac{W_{j,j+1}^{\text{UOD}}}{K_a A_j}$$

$$+ \frac{W_{j+1}^{UOD}}{A(K_a - K_1)} (e^{-K_1 \tau(x)} - e^{-K_a \tau(x)}) \quad (5)$$

The reaeration term, $(\phi_s - \phi_0)e^{-K_a \tau(x)}$, is redundant since it also appears in (4). Thus, in the actual operation of the model, this term is set to zero.

For lakes or reservoirs the dissolved oxygen, ϕ_L, is calculated according to the following formula, which assumes that the lake is in equilibrium and that oxidation takes place only in the top half of the lake:

$$\phi_L = \phi_s - \frac{C_L^{UOD} q_L}{K_a S_L} \quad (6)$$

C_L^{UOD} is the concentration of ultimate oxygen demand at the lake's input node, calculated by formulas (2) and (3) from both point and nonpoint sources of BOD_5 and TKN. The flow through the lake is q_L, and S_L is one half the lake's volume. The reaeration coefficient K_a has the value of about 0.1.

The final formulas are for estuaries. For each estuary, a maximum concentration of a pollutant is defined [*Hydroscience, Inc.*, 1971]. While this concentration occurs only in one region of the estuary (near its upper reach), it will be used as an 'index' of the quality of the entire estuary. The maximum concentration C^M is

$$C^M = \frac{W + C_0 q}{q[1 + (4K_1 E/U^2)]^{1/2}} \quad (7)$$

where W is the total direct waste load (in lbs/d) from both point and nonpoint sources, C_0 is the concentration of wastes at the node marking the entrance of the estuary, q is the flow at this entrance node, K_1, as before, is the temperature adjusted attentuation parameter, U is the advective velocity, defined as $U = V_L/20$, where V_L is the stream velocity at the entrance of the estuary, and E is the tidal dispersion coefficient. The tidal dispersion coefficient is defined by $E = 5.2(V_m)^{1.33}$, where V_m is the maximum tidal velocity (in knots) measured at a point about 10 miles from the beginning of the estuary [*Thomann*, 1972].

With K_1 set to zero, formula (7) would be suitable for conservative pollutants such as dissolved solids. However, because of the dominating effect of the ocean on ambient salt content, estimates of the concentration of dissolved solids will not be made.

A final use of formula (7) is to compute the estuary's maximum ultimate oxygen demand, UOD^M as a starting point to compute a maximum dissolved oxygen deficit, D_c. (In terms of earlier notation, $D_c = \phi_s - \phi$.) The dissolved oxygen deficit formula is

$$D_c = UOD^M \left\{\frac{X_3 - 1}{X_3 (X_1 - 1)}\right\} \left\{\frac{X_3}{X_2}\right\}^{(1/1-X_3)} \quad (8)$$

where

$$X_1 = \frac{K_a}{K_1} \qquad X_2 = \frac{(1 + 4X_1 X_4)^{1/2}}{(1 + 4X_4)^{1/2}}$$

and

$$X_3 = \frac{1 - (1 + 4X_1 X_4)^{1/2}}{1 - (1 + 4X_4)^{1/2}}$$

The parameter X_4 equals $K_1 E/U^2$ where K_1, E, and U are defined as before. This analysis is based on data from *Hydroscience, Inc.* [1971]. For estuaries only the decay rate K_1 does not depend on concentration but does depend on temperature. At 20° C, $(K_1)_{20} = 0.3$. Also, as before, $K_a = 7.6\ V/h^{1.33} 1.024^{(Temp-20)}$. The velocity V is set to one half the maximum tidal velocity V_m multiplied by 1.69 (to convert from knots to ft/s) and the depth h is equal to the depth of the entrance node of the estuary.

CONCLUDING COMMENTS

The conclusion as to whether this modeling structure and the associated assumptions provide adequate estimates of water quality must await further testing. Every opportunity will be taken to compare the model's estimates with available data on quality from such sources as EPA's STORET system and specific detailed river studies. Of course, in making such comparisons, care must be taken to make sure that the model's estimates and the other data are benchmarked to approximately the same pollution loadings.

In addition, the comparisons with monitored data should be made with an understanding of what is meant by adequate estimates of water quality. The adequacy of any model's outputs should be determined by the model's intended purposes. In our case we are interested in national assessments of environmental policy. Local and regional policy analyses may require much more precision and refinement than could be possibly provided by our model. Also, any comparison with monitored data should recognize the fact that these data have large, but often unreported, variances. Often apparent disparities between modeled estimates and the monitored data are actually statistically insignificant.

However, one particular deficiency of the model has come to light even before formal testing. In the uppermost reaches of the network, many small tributaries were neglected. While the hydrologic information for any node in these regions is reasonably accurate for the particular tributary on which the node is located, the flow of this node necessarily understates the total flow coming from all tributaries in the region. Because the pollutants from all counties in the region are assigned to the single node, the concentration of this node is consequently greatly overstated. (This problem becomes less serious for downstream nodes along larger branches and main stems, since these lower nodes necessarily 'capture' the flows from all upstream reaches.)

One possible solution to this problem would be to note any increase in flows between two successive nodes and then prorate some portion of this increase back to the upstream node and thereby increase its apparent flow. However, determining how large a portion to assign to the upstream node would be quite difficult. Moreover, the velocity, mean depth, and width parameters for this node would be inconsistent with the increased flow. These values, too, would have to be altered.

Such changes, while 'solving' the problem of estimating excessive concentrations, would create a distorted picture of the streamflow characteristics of the region. Certainly a better approach would be to increase the number of nodes for upstream portions of the network. Whether data problems preclude this is, at present, an open question. Project resources force us to postpone such an investigation until some future date.

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